Section 4.0 Passive Treatment Technologies

Introduction

Passive treatment encompasses a series of engineered treatment facilities that require very little to no maintenance once constructed and operational. Passive water treatment generally involves natural physical, biochemical, and geochemical actions and reactions, such as calcium carbonate dissolution, sulfate/iron reduction, bicarbonate alkalinity generation, metals oxidation and hydrolysis, and metals precipitation. The systems are commonly powered by existing water pressure created by differences in elevation between the discharge point and the treatment facilities.

Passive treatment does not meet the standard definition of a Best Management Practice (BMP). In general, BMPs consist of abatement, remediation, and/or prevention techniques that are conducted within the mining area (at the source) during active remining operations. Passive treatment, by its nature, is an end-of-the-pipe solution to acid mine drainage (AMD); it is treatment. These systems are frequently installed after reclamation to treat AMD. BMPs, on the other hand, are performed as part of the mining or reclamation process, generally not after the fact. If treatment, passive or conventional, is required for a discharge to meet effluent standards (BAT or some alternate standard), the operator is held liable and treatment continues, theoretically, until the discharges naturally meet the applicable effluent standards.

Regardless of whether or not passive treatment fits the definition of a BMP, it can be used as part of the overall abatement plan to reduce pollution loads discharging from remining sites. There are situations where passive treatment may be employed to improve water quality above what was accomplished by the BMPs. Therefore, a detailed discussion of the use of passive treatment technology to treat AMD in this manual is warranted. Passive treatment includes, but is not limited to:

- Anoxic limestone drains (ALDs)
- Constructed wetlands
- Successive alkalinity-producing systems (SAPS)
- Open limestone channel (OLCs)
- Oxic limestone drains (OLDs)
- Pyrolusite® systems
- Alkalinity-Producing Diversion wells

Passive treatment technologies also can be incorporated into the reclamation plan along with more traditional BMPs. For example, ALDs can be installed within the backfill as a type of pit floor drain. This has been done at a remining site on the Shaw Mines Complex in Somerset County, Pennsylvania, where an ALD 2,500 feet long, 30 feet wide, and 10 feet deep was installed within the backfill (Ziemkiewicz and Brant, 1997). Wetlands can be constructed where returning the site completely to the approximate original contour is not economical. Discharges can be routed through these wetlands for treatment. Open limestone channels can be used in the construction of diversion ditches or as pond outflow structures. Additionally, passive treatment can be employed on AMD-yielding discharges that would not otherwise be impacted by the operation or by integral BMPs. These discharges are hydrologically discrete from the operation.

Theory

Once installed, passive treatment systems require little maintenance through the projected life of the system. They are a low-cost method of treating mine water. However, these systems have a finite life and may require rebuilding or rejuvenation over the life of treatment. The period of treatment can be considerable; some mines have continually yielded AMD for well over a century. The power to run these systems is generated by changes in elevation that creates sufficient head and forces the water flow through them. The treatment is performed by natural, biological, geochemical, and physical actions.

Frequently, more than one type of passive treatment or an integrated system of passive treatment technologies are employed to treat mine drainage. These facilities, like conventional treatment facilities, are typically designed to raise the pH and remove metals (e.g., iron, manganese, and aluminum) of acid mine drainage.

Site Assessment

In order to determine the feasibility of integrating passive treatment into a remining operation BMP plan, there are several factors that need to be assessed. The most critical is the determination of the water quality and discharge rates. These data need to be collected and analyzed on a seasonal basis to completely characterize discharge(s). Sampling at least once per month, for a complete year, is recommended. Additional monitoring may be required, if the precipitation has been substantially above or below normal. These data directly relate to the sizing of passive systems.

Of particular importance in selecting the type of passive treatment system(s) is the water quality characteristics of the discharge. Dissolved oxygen (DO) concentration in the water as it emanates, speciation of the dissolved iron (i.e. ferrous and ferric) concentrations, dissolved aluminum concentration, net acidity or alkalinity, and pH are all important parameters. The concentrations of dissolved manganese and sulfate are of lesser importance (less problematic), but should also be determined.

Determination of the discharge flow rate is perhaps the most critical data for the sizing and selection of passive treatment technologies. Without accurate flow data, an improperly sized passive treatment system may either under treat the water or be much larger, and thus more expensive, than needed. Flow measurements should be determined at the time water samples are collected and should be performed using standard scientifically accepted means. A weir (e.g., v-notch) or flume (e.g., H-type), timed-volumetric (e.g., bucket and stopwatch), or flow meter and cross sectional area are acceptable and commonly used methods to determine flow. It is recommended that at least one extreme high flow and low flow be sampled during the monitoring period. If the flow is too low or too erratic, some types of passive treatment (e.g., wetlands, SAPS) may not be suitable.

Most passive treatment systems require a sufficient gradient to create the desired head to drive the water flow through the treatment systems. Therefore, implementation of these systems requires a large enough area for construction sufficiently down gradient of the discharge.

4.1 Implementation Guidelines

Anoxic Limestone Drains

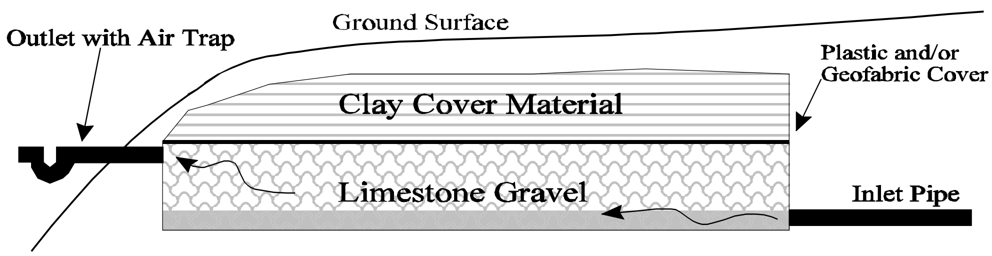
In general, attempts to use limestone to treat acidic ferruginous mine drainage at the ground surface commonly fail after a short time period. These failures are caused by the low dissolution rate of limestone at atmospheric levels of CO_2 and by iron (ferric) hydroxide ($FeOH_3$) armoring of the limestone. Limestone armoring virtually halts all bicarbonate alkalinity production from the dissolution of calcium carbonate. Once exposed to the atmosphere, the iron in mine drainage rapidly oxidizes from ferrous (Fe^{2+}) to the ferric state (Fe^{3+}). Once oxidized, the ferric iron will quickly precipitate out of solution, coating the limestone, and creating an iron hydroxide precipitate sludge known as "yellow boy". However, if mine drainage is maintained in a low oxygen (anoxic) environment, the iron will remain in the ferrous state and will not readily precipitate from solution. Anoxic mine water passing through limestone drains allows for the production of alkalinity without iron armoring and precipitation. For these drains to function properly, the mine drainage dissolved oxygen content should be less than 1 mg/L (Kepler and McCleary, 1994). Cravotta (1998) states that dissolved oxygen in the water should be less than 0.3 mg/L to preclude iron oxidation.

Anoxic limestone drains are designed to generate alkalinity in acid mine drainage without atmospheric exposure. In addition to preventing iron hydroxide precipitation, the closed environment of an ALD fosters increased CO_2 concentrations, which in turn facilitates higher alkalinity production. Alkalinity production in ALDs is much greater than what can be expected at atmospheric CO_2 levels. CO_2 partial pressures ranging from 0.022 to 0.268 atmospheres were calculated for 21 ALDs (Hedin and Watzlaf, 1994). The production of 61 mg/L alkalinity under atmospheric conditions can quickly be increased to over 450 mg/L within an ALD (Hem, 1989; Hedin and Watzlaf, 1994). The mechanism for the increased alkalinity production from higher CO_2 concentrations is discussed in Section 2.0 and 3.0. Removal of acidity from mine water flowing through ALDs ranges from 0 to over 5900 mg/L. The higher levels of acidity removal are attributed to loss of mineral acidity from detention of ferric iron and aluminum within the drains. This detention of ferrous iron was observed at two sites using ALDs with the detention times exceeding 25 hours (Hedin and Watzlaf, 1994). The lower acidity and higher alkalinity of the water once it leaves the drain cause the pH of the water to rise, which in turn significantly increases the precipitation rate of iron and other metals.

ALDs are often installed to aid the efficiency of constructed wetlands. These wetlands work more effectively to remove metals if the pH of the water is raised by ALD pretreatment. Most metals associated with AMD will precipitate more readily from solution in a high pH environment. Nairn and others (1991) stated that a pH of 6.0 (standard units) and a net alkalinity allow passive treatment systems (constructed wetlands and settling ponds) to work much more effectively.

Design and construction of an ALD should be based on the required detention time for the maximum flow anticipated for the discharge over the effective life of the facility. The discharge water quality should also be considered. It is recommended that an environmental safety factor be employed in design to cover the worst case scenario. The discharges should be monitored for at least one year prior to system installation to determine the range of flows anticipated and the variability of water quality. Precipitation records during the monitoring period should be compared to average years to determine the representativeness of the flow and water quality data. Configuration and size of ALDs are based on the flow rate, projected life of the system, purity of the limestone, and desired water quality. The ALD should be able to treat the water to the desired levels under all flow conditions. Design details of ALDs can vary, but the general configuration is relatively consistent. Figure 4.1a illustrates the basic construction of an ALD.

Figure 4.1a: Typical Anoxic Limestone Drain Construction



Hedin and Watzlaf (1994) analyzed water quality and flow data from 21 completed ALDs treating AMD in Appalachia to determine their efficiency. They determined that an in-drain detention time of at least 15 hours and perhaps as high as 23 hours is required to produce the maximum alkalinity. ALD sizing criteria were developed based on the discharge rate, a minimum 15 hour detention time, the desired life of the drain, and physical and chemical properties of the limestone used. The equation derived is as follows:

$$M = \frac{Qp_b t_d}{V_v} + \frac{QCT}{x} \qquad \text{(Equation 1)}$$

Where:
M = mass of the limestone
Q = discharge rate
p_b = bulk density of the limestone
t_d = the detention time
V_v = bulk void volume expressed as a decimal (20 percent voids is expressed as 0.20)
C = predicted concentration of alkalinity of drain effluent
T = designed life of drains in years
x = calcium carbonate content of the limestone in decimal form

An example calculation of drain size in metric tonnes (mt) is as follows. The calculation assumes a discharge rate of 30 L/min, limestone bulk density of 1600 kg/m^3, bulk void volume of 40 percent, a projected alkalinity of 300 mg/L, a limestone calcium carbonate content of 95 percent, and a life of 25 years.

$$M = \frac{(30\ \mathrm{L/min} \times 60\ \mathrm{min/hr})\ (1600\ \mathrm{kg/m^3} \times \mathrm{m^3}/1000\mathrm{L} \times \mathrm{mt}/1000\ \mathrm{kg})\ (15\ \mathrm{hr})}{0.40}$$

$$+ \frac{(30\ \mathrm{L/min} \times 60\ \mathrm{min/hr})\ (300\ \mathrm{mg/L} \times \mathrm{mt}/109\ \mathrm{mg})\ (25\mathrm{yr} \times 8766\ \mathrm{hr/yr})}{0.95} = 232.6\mathrm{mt}$$

ALDs are located down gradient of the discharge point to allow for a free-flowing, gravity-driven system. A sufficiently wide and deep trench is dug to accommodate the amount of limestone needed to provide the desired detention time to yield the maximum alkalinity. Dimensions of ALDs commonly range from 2 to 9 feet wide and 150 to 1500 feet long; however, much larger drains have been constructed. Drain depth should be enough to hold a 2 to 6 feet thick layer of limestone with sufficient cover to preclude infiltration of oxygen (Nairn and others, 1991). Once excavated, the trench is filled with crushed limestone. Brodie and others (1991) recommended that the size of the limestone be 0.75 to 1.5 inches to give both the needed surface area and needed drain hydraulic conductivity. Purity of the limestone should be as high as possible to prolong the functional life. Use of a low-purity limestone would require the drain to be larger and more limestone material to be used.

Mine drainage is piped into the ALD directly from the source, before it has been exposed to the atmosphere. It is common to dig into the discharge point and install a buried collection and piping system. The drain inlet is usually at the base of the drain to maximize limestone contact. The limestone is covered with 10 to 20 mil (0.01 to 0.02 inches) thick sheet plastic followed by geosynthetic fabric to prevent puncturing of the plastic. The fabric is then covered with lightly compacted clay. The plastic and clay are emplaced to inhibit the infiltration of atmospheric oxygen. Clay is then covered with soil. The clay and soil should be at least 2 feet thick to effectively prevent oxygen infiltration. The surface should be crowned (mounded) to inhibit erosion and water infiltration and to accommodate long-term subsidence as the limestone dissolves. Brodie and others (1991) recommend that the drain should be rip rapped or vegetated with a plant species that will discourage the growth of trees, such as sericea or crown vetch. Tree roots could breach the drain seal and allow oxygen infiltration. The outflow pipe is installed at the top of the limestone trench opposite to the inflow point. The outflow pipe is equipped with an air trap to prevent oxygen migration into the drain. The elevation of the outflow pipe should be below the head elevation driving water through the drain. The inflow and outflow piping size should be large enough to permit unrestricted flow for the highest projected discharge rates.

Once the water exits the drain and is subaerially exposed, dissolved iron and most other dissolved metals in the water will rapidly oxidize and begin to precipitate out. It is recommended that the water be diverted to a settling basin or pond sized for this purpose. The settling basin will greatly extend the life of a constructed wetland or other subsequent treatment facility. Ideally, the alkalinity yielded by the drain will be high enough to neutralize the existing mine water acidity as it enters the drain and to neutralize the mineral acidity created subsequently by the oxidation and hydrolysis of the iron and metals after the water exits the drain.

There are some restrictions to using ALDs to treat AMD. Most are related to the mine water quality. If the dissolved iron in the discharge water has been oxidized to the ferric state prior to entering the drain, the drain will eventually fail. Ferric iron will readily precipitate in the drain once the pH of the water is sufficiently raised, armoring the limestone and clogging the void spaces. This precipitation decreases the drain efficiency and eventually causes failure in terms of limestone dissolution rate and/or water not flowing through the drain. The introduction of DO to the mine water will allow iron oxidation to the ferric state. Therefore, the available atmospheric oxygen should be restricted. These drains are not recommended to treat mine water with high concentrations of dissolved aluminum, because aluminum will also precipitate out in the drain once the pH is raised with or without oxidation. It is not recommended to use a dolomitic limestone, because the dissolution rate of dolomite ($CaMg(CO_3)_2$) is much slower than calcium carbonate. Therefore, the effectiveness of the drain would be diminished or the drain size would have to be increased to accommodate the lower reaction rates. If sulfate concentrations exceed 2000 mg/L, it is possible for gypsum ($CaSO_4 + 2H_2O$) to precipitate within the drain once the pH is raised and calcium concentration is increased (Ziemkiewicz and others, 1994).

Constructed Wetlands

The possibility of using constructed wetlands to treat AMD was first indicated by observations made on the treatment of mine drainage by naturally-existing wetlands. The flow of AMD through *Sphagnum* moss bogs illustrated that iron and acidity concentrations could be reduced without degrading the wetland. Studies on naturally-formed wetlands treating mine drainage were initially conducted in Ohio and West Virginia. Both studies showed that iron and acidity were substantially decreased and the pH of the water was raised after flowing through the wetlands (Kleinmann, 1985).

Because of the beneficial effects observed at natural wetlands, numerous wetlands have been constructed in attempts to treat acid mine waters passively. *Sphagnum* moss was used initially because it was observed to be successful in natural wetlands and preliminary studies showed that it can remove large quantities of iron (Kleinmann, 1985). Near surface oxidation and sulfate reduction in deeper organic-rich zones also decrease the amount of iron in wetlands. Later, cattail (*Typha*) wetlands were constructed to treat mine drainage. This change in vegetation appears to be related to limited iron detention from cation exchange by *Sphagnum* moss and the high sensitivity of the moss to wetland water levels. Studies showed that most of the iron detention in constructed wetlands was due to binding to the organic matter and the direct precipitation of iron hydroxides (Wieder, 1988).

There are two ways that constructed wetlands treat AMD. First, aerobic reactions cause oxidation and hydrolysis of the metals forming metal hydroxide precipitates. This removal of metals has a tendency to release mineral acidity and lower the pH of the water. Aerobic wetlands work primarily with mine water flowing through at or very near the surface. The subaerial exposure permits oxidation of iron and other metals. However, in order for these wetlands to work most efficiently, the water needs to have a pH of 6.0 or higher and a net alkalinity. At a pH of 6.0 or higher, the rate of iron oxidation dramatically increases. At pH levels below 6.0, manganese oxidation virtually halts. As these metals oxidize and hydrolyze, mineral acidity is released and the pH will decrease. Therefore, the more efficient wetland systems will have an excess net alkalinity in the water prior to the precipitation of the metals to buffer (the ability to hold the pH relatively steady with the addition of an acid or a base) the release of mineral acidity.

Second, anaerobic reactions that occur under anoxic conditions cause sulfate reduction. Under anaerobic conditions, metals are removed in reduced forms (metal sulfides), and bicarbonate alkalinity is created. Anaerobic wetlands, also called compost wetlands, support reducing conditions within the substrate. Sulfate reduction by sulfate reducing bacteria (e.g., *Desulfovibrio* and *Desulfomaculatum*) is one of the primary anaerobic reactions (Smith, 1982). Sulfate-reducing bacteria thrive in anoxic environments, feed on organic material, and utilize sulfate in their respiration processes. The organic substrate acts as an oxygen sink in natural and constructed wetlands, creating suboxic or anoxic conditions from the bacterial decomposition of the organic matter. Oxygen in water flowing through the organic substrate is rapidly removed. With sulfate reduction, hydrogen sulfide gas (H_2S) is created and a variety of metal sulfides (e.g., pyrite (FeS_2), iron monosulfides (FeS)) are formed and deposited within the substrate. Wetland flow systems designed to force water through the organic substrate promote sulfate reduction on a larger scale. In the process of sulfate reduction, bacteria use organic carbon (CH_2O) and sulfate (SO_4^{2-}), producing hydrogen sulfide (H_2S) and bicarbonate alkalinity (HCO_3^-) (McIntire and Edenborn, 1990) as shown in Equation 2. The production of bicarbonate alkalinity neutralizes acidity and raises the pH of the water.

$$2CH_2O + SO_4^{2-} = H_2S + 2HCO_3^- \qquad \text{(Equation 2)}$$

There are a multitude of configurations for constructed wetlands. However, a few researchers have developed criteria for wetlands sizing and design to maximize AMD treatment. Kleinmann (1985) suggested that 200 ft^3 of wetland are required for each gallon per minute of discharge.

Kleinmann indicated that constructed wetlands may be most applicable to discharges of no more than 10 gpm, a pH over 4.0, and iron concentration of 50 mg/L or less. Attempted uses of wetlands to treat discharges with water quality or quantity exceeding those criteria were mostly unsuccessful.

Hedin and Nairn (1990) determined that loading (mass/time) directly related to the wetland treatment area was a more appropriate criteria for wetland engineering. They developed a sizing formula based on iron grams per day per meter squared (Fe g/day/m^2 or gdm) of wetland area. The method also factored in pH, flow, and iron concentration. A sizing criterion of 10 gdm of iron was determined for water with a pH of 4.0. For water with a pH of 3.0, the efficiency drops to 4 gdm of iron.

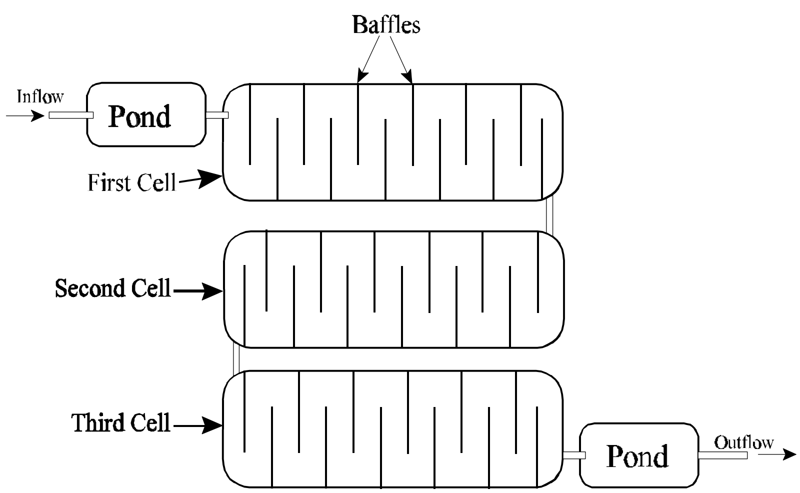
Kepler (1990) observed that there may be other factors that also play a role in the efficiency of wetlands to treat mine water. He noted seasonal variations in the treatment effectiveness related to variations in influent iron loadings as well as treatment area and biological efficiency. An inverse relationship was observed between the iron load (ferrous and ferric iron ratio) and the efficiency of the wetland. This is related to the flow system through the wetland allowing time for aerobic and anaerobic reactions to occur. He indicated that the flow system may be as important as the surface area or vegetation types. For overall effectiveness, a value of 15 gdm was determined for year round treatment. A sizing safety factor of 1.25 was also recommended (Kepler, 1990).

Stark and others (1990) in a study of a *Typha* wetland near Coshocton, Ohio, observed a consistent treatment efficiency at 10 gdm. However, the site averaged over 13.5 gdm. They likewise recommended that wetlands be sized to treat the maximum loads anticipated.

It is critical that accurate discharge flow and water quality background data are collected for at least one water year (October 1st through September 30th). Extreme care should be taken to ensure that flows are accurately measured. Wetlands should be sized for the maximum forecasted flow, concentration, and load, so extreme conditions can be successfully treated.

Although configuration of constructed wetlands can vary widely, there are some basic common components. Figure 4.1b is a schematic diagram of a typically constructed wetland system. In many instances, the mine discharge is initially diverted to a small settling pond. Depending on the pH and alkalinity of the water, some iron will precipitate within the pond, extending the working life of the wetland. The water then flows from the pond into a large wetland cell or series of cells. The water course is designed so the detention time is as long as possible to yield maximum treatment. This is usually accomplished by the inclusion of a series of baffles to divert the water along a circuitous path. The last wetland cell is followed by a final "polishing" pond to allow for precipitation of any appreciable remaining iron. After the final pond, the water, if meeting applicable effluent standards, is discharged to the receiving stream. If effluent standards are not being met, additional treatment may be required.

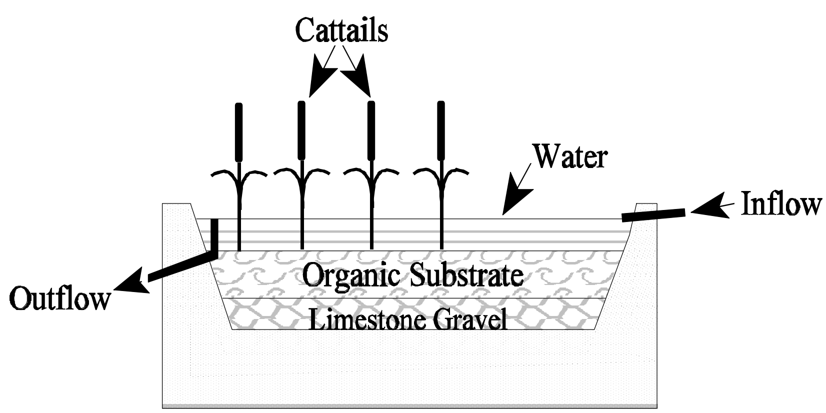
Figure 4.1b: Commonly Constructed Wetland Diagram



Construction design of individual wetland cells is directly dependent on the amount of flow and water chemistry. Brodie and others (1988a) based the size and number of cells on the projected flow from a 10-year, 24-hour storm event. The cell size is based on the area required to treat the flow for iron concentration, according to grams/day/m^2 of iron, as discussed above. Cell dimensions are based on the treatment area needed, maximization of the flow path, site topography, and configuration of the available space down gradient of the discharge.

Wetland cells are frequently lined with an initial thin layer of crushed limestone that is usually about 6 inches thick (Figure 4.1c). The limestone is covered with a thicker organic layer, usually 12 to 18 inches. Mushroom compost is the most common material used for the organic substrate. The cell is subsequently flooded with 6 to 12 inches (15 to 30 cm) of water and planted with vegetation. Cattails are by far the most commonly planted vegetation in constructed wetlands. Other plants used include, but are not limited to, cattail-rice cutgrass, sphagnum moss, rushes, and bulrushes (Brodie, 1990; Brodie and others, 1988b). Various types of blue-green algae (Cyanobacteria) have also been introduced into wetlands in attempts to improve efficiency for manganese reduction (Spratt and Wieder, 1988).

Figure 4.1c: Typical Wetland Cell Cross Section



There are limits to which wetlands can be used to treat mine water. One of the most salient problems is the amount of area required. A high-flowing, high-iron discharge requires a huge area for treatment. A low pH (<4.0) water will require more treatment sizing (4 gdm) than a higher pH (>4.0) water (10 gdm). Using the sizing criteria developed by Hedin and Nairn

(1990), a mine discharge of 600 gpm, 75 mg/L of iron, and a pH greater than 4.0 would require a wetland area of at least 6.1 acres and an area of 15 acres for a pH under 4.0. However, Hedin and Nairn (1990) stated that for "highly contaminated drainage," a larger wetland sizing criterion may be required. At a pH of 3.0, the wetland sizing may need to be increased by 300 percent.

The performance of aerobic wetlands is greatly hampered by low-pH water. Raising the pH prior to piping the water into the wetland will greatly improve iron removal. ALDs have been used successfully in conjunction with wetland treatment. The increased alkalinity buffers the decrease in pH caused by release of mineral acidity from iron hydrolysis. This buffering in turn improves the treatment ability of the wetland (Brodie and others, 1991).

By design, iron hydroxide will precipitate within constructed wetlands. This precipitation will eventually cause iron hydroxide sludge buildup in the cells, which will cause changes to the water levels. These changes will adversely impact the vegetation and decrease the wetland treatment ability. Also organic material will eventually be depleted through bacterial action, and require replacement. Depending on the flow system, the limestone may also need to be replenished as dissolution occurs. Therefore, over time, wetlands require periodic maintenance to remove the iron hydroxide sludge and replace substrate materials.

Successive Alkalinity-Producing Systems

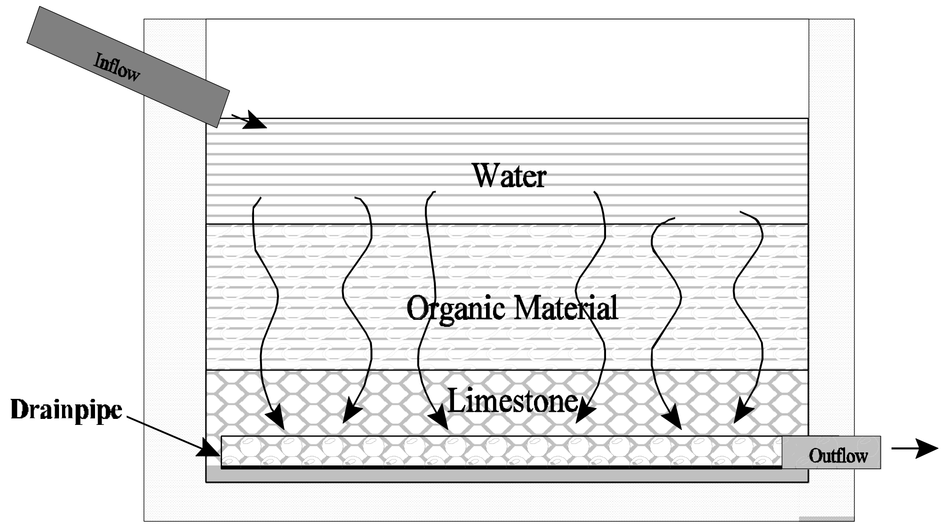
Successive alkalinity-producing systems utilize the alkalinity production of anaerobic wetlands and ALDs to remove metals from mine water, while greatly increasing the alkalinity production over either of the two systems working singly. With SAPS, the ALD criteria for anoxic mine water and the requirement of ferrous iron does not apply. An oxygen sink is created by anaerobic sulfate reduction which will reduce any ferric iron (Fe^{3+}) to ferrous.

Construction of individual SAPS cells is similar to that of a constructed wetland cell, but the flow system differs and no vegetation is required. Because SAPS work on the concept of a series of steps that produce alkalinity, there are several configurations for the entire system. Kepler and

McCleary (1994) suggested a configuration of an ALD followed by an aerobic wetland or settling pond, which is then followed by a SAPS cell that discharges into a second aerobic wetland or settling pond.

An individual SAPS cell is designed to accept water inflow at the surface and drain from the bottom. The basal layer in a SAPS cell is crushed limestone covering perforated underdrain pipes (Figure 4.1d). Skousen and others (1995) suggested that the underdrain pipes be covered with 12 to 24 inches of limestone. However, Kepler and McCleary (1994) indicate that the thickness of the limestone layer is based on the detention time required for maximum alkalinity production. A similar amount of detention time as that required for an ALD is recommended. Four SAPS constructed in Pennsylvania had limestone layers ranging in thickness from 18 to 24 inches (Kepler and McCleary, 1995). A layer of organic matter, usually mushroom compost, is placed over the limestone. The thickness of the organic layer, like the limestone layer, is based to a large extent on the required detention time. Kepler and McCleary (1995) observed four sites in Pennsylvania where the organic layers were 18 inches thick. Skousen and others (1995) recommended 12 to 18 inches of organic material. Overlying the organic layer is free-standing mine water. The depth of the water is dependent on the head (pressure) required to drive the water through the organic and limestone layers at a rate that to adequately achieve the required the biochemical and chemical reactions (discussed below). Kepler and McCleary (1995) indicated a depth range of 5.25 to 6.23 feet was adequate at the four study sites in Pennsylvania; whereas, Skousen and others (1995) suggested a water depth of 4 to 8 feet. Size of the SAPS is based on the required water detention time, which is related to the flow rate, more so than the water quality. The rate of atmospheric oxygen diffusion into a body of water is relatively constant and should be used in determining the areal size of the SAPS cell.

Figure 4.1d: Example of a Successive Alkalinity-Producing System Cell



SAPS function through a series of chemical and biochemical reactions to remove iron and other metals from the water, while increasing the alkalinity. When mine water is initially discharged into the SAPS cell, it does not matter if the water has been oxygenated or the iron has been oxidized to the ferric state. Some of these metals, especially iron, will oxidize in the shallower water and precipitate on top of the organic layer. Kepler and McCleary (1994) observed 2 inches (5 cm) of iron hydroxide deposited in a SAPS at a mine site in northwestern Pennsylvania.

Once in the cell, the water flows downward toward the organic layer and the water is rapidly stripped of dissolved oxygen by microbial decomposition of the organic material. Bacteria utilize the DO in the mine water to metabolize the organics. These reactions occur near the interface of the organic material and the water. Kepler and McCleary (1994) reported that water nearly saturated with dissolved oxygen (~10 mg/L) entering the cell was virtually anoxic (<0.2 mg/L) after passing through the system. Oxygen can only infiltrate several centimeters into the organic substrate (Kepler and McCleary, 1994). Once the dissolved oxygen is removed, anaerobic sulfate-reducing bacteria in the organic layer will chemically reduce the metals as well as the sulfate ions, yielding hydrogen sulfide (H_2S) gas and metal sulfides. The H_2S will be released into the atmosphere, where it subsequently oxidizes to form water and native sulfur (S) (Lehr and others, 1980). When these systems are working properly, considerable H_2S is yielded and the systems tend to have an offensive smell. H_2S smells similar to rotten eggs and is unpleasant even at very low concentrations (0.05 mg/L). Metal sulfides are deposited within the organic material, but some of the reduced metals will remain dissolved and pass through the organic layer.

This reduction process also yields bicarbonate alkalinity to the water as described in the preceding wetlands section. This process, in turn, will neutralize acidity, add alkalinity, and raise the pH of the water.

Once the water has passed through the organic layer, it enters the underlying limestone gravel. Because the oxygen has been stripped from the water, and any metals that are not precipitated are in a reduced state, the limestone layer functions as an ALD. Passage through the limestone adds additional alkalinity to the water through dissolution of the calcium carbonate, as described above under ALDs. If the SAPS are properly sized, the effluent should have a pH of 6.0 or higher (Skousen and others, 1995). Aluminum tends to pass through the organic layer and is precipitated in the limestone. Because aluminum precipitate does not armor the limestone, but instead remains as loose precipitate, it can eventually plug the limestone layer. Therefore, a piping system that will allow a periodic forced flushing of the limestone layer is needed to maintain the efficiency of the system (Kepler and McCleary, 1997).

The SAPS cell effluent is typically piped into a conventional aerobic wetland or settling pond. With the excess alkalinity yielded by the SAPS, much of the remaining metals (mainly iron) will quickly precipitate out of solution in the wetland or pond. The process of iron oxidation and hydrolysis will, as discussed earlier, yield acidity. However, the excess alkalinity in water from a well-designed SAPS should perform a buffering action and be sufficient to maintain a net alkalinity throughout this secondary precipitation process. If the alkalinity is insufficient to neutralize the acidity produced by the iron precipitation, the water can be piped through a second SAPS. This process can be repeated until the mine water meets the applicable effluent standards.

Limitations on SAPS construction, use, and maintenance are similar to those for wetlands and ALDs. Restrictions to the use of SAPS include, but are not limited to:

- Engineering and sizing should be determined by the discharge flow rate. The highest anticipated flow rates should be used as an engineering guideline.
- Topography should be such that the system will function (flow) properly without the need for additional power.
- The organic material and limestone will eventually be exhausted and will need to be replaced.
- The water level needs to be deep enough that significant continued diffusion of dissolved oxygen at depth is prevented.
- There should be some mechanism to control the water level in the SAPS cell. This is important during extremely low flow periods, because the organic material could be subaerially exposed and dry out, thus shutting down oxygen removal and sulfate reduction. At high flows, the system could be overwhelmed.
- Iron sludge can eventually fill the pre- and post-SAPS ponds and will require periodic cleaning. If the iron precipitation within the SAPS is substantial, this will also require a periodic cleaning.
- Calcium carbonate purity of the limestone should be the highest available to prolong the life and maximize alkalinity production.
- Aluminum tends to precipitate in the limestone layer just as with ALDs. Therefore, a system is required to permit periodic flushing of the aluminum floc from the limestone.

Open Limestone Channels

In contrast to treating AMD with limestone in an anoxic environment, more recent research has been conducted on this treatment in an environment open to the atmosphere (oxic). As previously stated, when dissolved iron is oxidized, it will precipitate, armoring limestone and creating an iron hydroxide sludge. In theory, limestone, even if completely armored with iron, will continue to yield some alkalinity. Ziemkiewicz and others (1994) indicated that $CaCO_3$ in fully armored limestone is 20 percent as soluble as that in unarmored limestone. However, Ziemkiewicz and others (1996) reported that armored limestone may exhibit 25 to 33 percent of the $CaCO_3$ solubility of unarmored limestone. They observed an acidity reduction of 0.029 to 1.77 percent per foot of open limestone channel (OLC). Though rapid neutralization of acidity by armored limestone is observed initially, it slows with time, and exhibits a logarithmic decay of the neutralization rate (Ziemkiewicz and others, 1996).

Limestone channels are sized based on a projected 90 percent acidity neutralization with one hour of contact time or 100 percent acidity neutralization with three hours of contact time. Construction criteria are determined from the flow rate, channel slope, and acidity concentration. This information will determine the mass of limestone, the cross-sectional area and length of the drain, and ultimately, the in-channel detention time. Channels are constructed with an initial dam-like structure at the up-stream end to trap sediment and other debris and keep it from clogging the pore spaces between the limestone material throughout the remainder of the channel (Ziemkiewicz and others, 1994). OLCs also require sufficient slope, hence water velocity, to prevent clogging of the interstitial pore spaces with iron, manganese, and aluminum floc. If the pore spaces are substantially filled with metal floc, the water will flow over the top and be precluded from contacting the armored limestone, greatly attenuating, if not eliminating predicted dissolution rates.

Table 4.1 presents examples of limestone tonnage calculated to treat mine drainage with 1000 mg/L acidity, in an OLC with a cross section 3 feet deep by 10 feet wide. A mine discharge of 200 gpm and 1000 mg/L acidity would require a channel 3 feet deep, 10 feet wide, and 401 feet long filled with 5,085 tons of armored limestone to treat 100 percent of the acidity.

Table 4.1: OLC Sizing Calculations

	Channel Length in feet		Tons of Limestone Required			
			100% Dissolution		20% Dissolution	
Flow in gpm	1 hour contact time	3 hour contact time	1 hour, 90% Treatment	3 hour, 100% Treatment	1 hour, 90% Treatment	3 hour, 100% Treatment
100	67	201	169	508	847	2,542
200	134	401	339	1,017	1,695	5,085
500	334	1003	847	2,542	4,237	12,712
1000	669	2006	1,695	5,085	8,475	25,424

Modified after Ziemkiewicz and others (1994)

A recommended size of limestone gravel for use in these channels is greater than 4 inches in diameter (Ziemkiewicz and others, 1994). Optimal efficiency may be reached with limestone in the 6 to 12 inch diameter range. A channel grade exceeding 10 percent is also recommended to facilitate flushing of the metal floc from the drain, preventing a clogging of the pore spaces. Channels with less than a 9 percent grade were shown to be much less effective than channels with steeper grades (Ziemkiewicz and others, 1996). Because these channels are designed to flush out the metal floc, settling ponds are often constructed at the outlet point. These ponds will allow the metal floc to be concentrated at one point and should permit discharging the compliance water to the receiving stream. However, ponds will require periodic cleaning to maintain efficiency.

Open limestone channels are relatively simple and inexpensive systems to construct. However, there are some limitations to their use. Neutralization ability of these channels is greatly limited by the dissolution rate of armored limestone, atmospheric CO_2 concentrations, and contact time. Additionally, the reported dissolution rates (Ziemkiewicz and others, 1994; 1996) may be greater than what is chemically possible. Acidity reduction of up to 5 percent may occur due the formation of the minerals swartzmanite and jarosite, which store acidity (H^+). Calcium concentrations indicate the limestone dissolves at a rate considerably below 5 percent (Rose, 1999). In order to treat relatively large discharges with considerable acidity concentrations, very long drains (>3000 feet) with thousands of tons of limestone would be required. Therefore, these channels may not be applicable to space-limited mine sites. These channels require at least a 10 percent slope to prevent clogging, so they cannot be constructed in areas without the required topography or where the receiving stream is too near.

Oxic Limestone Drains

An oxic limestone drain, unlike an ALD, is designed to treat water containing appreciable dissolved oxygen and iron that has been oxidized (ferric). Like ALDs, OLDs are designed to promote higher limestone dissolution, hence alkalinity production, by concentrating the partial pressure of CO_2 (Pco_2). The Pco_2 is increased because the drain is covered, hampering its escape. The limestone dissolves rapidly enough to make the surface an unstable substrate for iron armoring, because the chemical reactions within the drain cause the dissolution of 2 moles of $CaCO_3$ for each mole of $Fe(OH)_3$ produced. The iron hydroxide ($Fe(OH)_3$) and aluminum hydroxide ($Al(OH)_3$) will precipitate to some extent within the drain. However, Cravotta (1998) observed that some of the metal flocs were "loosely bound" and were eventually carried down through the drain with water velocities 0.33 to 1.31 feet per minute and residence times $\leq$3.1 hours (Cravotta and Trahan, 1999). Additionally, the drains can be designed for periodic flushing to preclude buildup of these metal hydroxides.

There has been limited research on the use of OLDs to treat mine drainage. AMD with a moderate acidity concentration (< 90 mg/L), a pH of less than 4.0, and moderately low dissolved metal (iron, manganese, and aluminum) concentrations (1 to 5 mg/L) was treated using an OLDs (Cravotta, 1998).

The drains studied exhibited decreased iron and aluminum concentrations of up to 95 percent. Initially (first 6 months), manganese concentrations were unaffected by the drains. After the initial 6 months, the manganese concentrations were lowered by 50 percent, because of coprecipitation with the $Fe(OH)_3$ facilitated by higher pH (>5.0) of the water. The higher pH was due to increased alkalinity production as the water flowed through the drain. The rate of alkalinity production was greatest initially and decreased as the water traveled through the drain (Cravotta, 1998). This observation was likely caused by the more aggressive nature of the water as acidity (H^+) is released with the formation of $Fe(OH)_3$.

Drain sizing criteria are based largely on the discharge rate and desired alkalinity production. The discharge rate relates to in-drain residence time, which in turn is related to treatment effectiveness. Cravotta (1998) recommends that a perforated-pipe under drain be installed to permit periodic flushing of the precipitated metal hydroxides.

Although the research and use of OLDs are limited at this time, these drains may be a low cost method of treating low-level mine drainage. These drains will likely fail to effectively treat if:

- The flow rates are too high for the required detention time.
- The acidity is higher than the limited reaction rates allowed by the drain .
- The metal concentrations of the inflowing water are well above those previously tested.
- Drain clogging cannot be prevented or abated.
- The Pco_2 cannot be maintained at a high level.

The Pyrolusite® Process

Manganese removal from AMD is extremely difficult and has been historically costly. Manganese does not precipitate as easily as iron, and certain manganese oxides are soluble in the presence of ferrous iron. For these reasons, many operators should raise the pH to above 10 in order to effectively precipitate it out of solution (Kleinmann and others, 1985). The elevated pH then becomes problematic, because it is out of compliance (6.0 to 9.0 standard units) and extremely costly in terms of reagent and facility sizing. The manganese effluent standards were originally established as a surrogate rather than establishing standards for a series of toxic metals at mine treatment facilities, to some extent due to the detriment of manganese on the stream quality, and the best practicable control technology (BPT) of existing water treatment facilities (Kleinmann and Watzlaf, 1986). However, the toxicity of manganese on aquatic life has not been conclusively established. An effective and inexpensive passive method to treat manganese in AMD has been actively pursued for several years.

Vail and Riley (1997) reported on a biologically-driven patented process to remove iron and especially manganese from mine drainage, while raising the alkalinity of the water. In this process, a bed of crushed limestone is inoculated with "cultured microorganisms" that oxidize iron and manganese in the water contacting the bed. These aerobic microorganisms produce relatively "insoluble metal oxides" while yielding alkalinity by "etching" the limestone hosting medium. The microorganisms are environmentally safe and are not biologically engineered (Vail and Riley, 1997). The metal oxides formed during this process are believed to be manganese dioxide or pyrolusite (MnO_2) and hematite (Fe_2O_3). Both metal oxides are relatively stable and insoluble in alkaline water.

The system is designed so that the water has a protracted contact with the limestone with a recommended minimum residence time of 2.5 to 3.0 days. The engineered treatment cell size should be based on a projected maximum peak flow. The purity of the limestone should be at 87 percent $CaCO_3$ or greater (Vail and Riley, 1997). The hydraulics of the cell should be managed to maximize water contact with the limestone substrate.

Results from a Pyrolusite® process cell monitored over a 5 year period showed a dramatic reduction in metals and an increase in the pH. An average influent of 30 mg/L manganese was reduced to below 0.05 mg/L in the effluent. Inflowing iron ranged from near 1 to over 115 mg/L, while the effluent was consistently below 1 mg/L. The pH of the water was raised over 2 orders of magnitude from about 4.5 to over 7.0. The pH improvement is directly attributable to a dramatic increase in the alkalinity from about 10 mg/L or less to an average of nearly 80 mg/L (Vail and Riley, 1997).

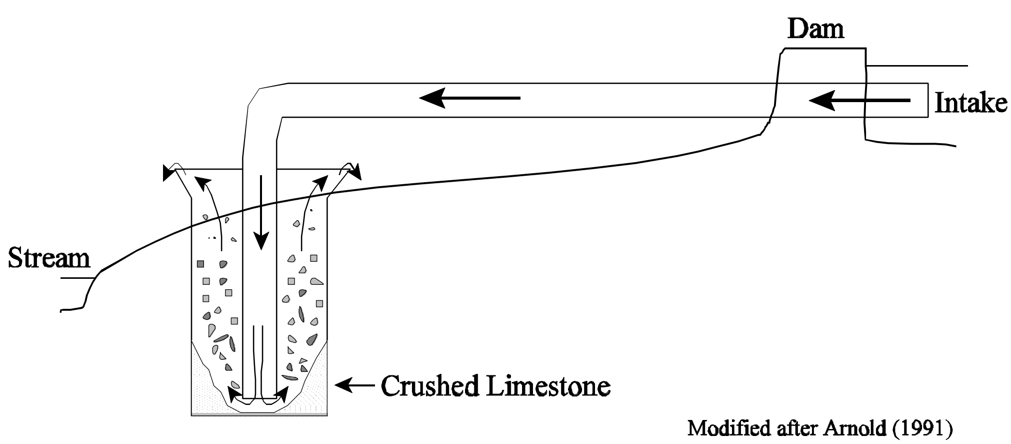
Restrictions on the use of Pyrolusite® cells stem to some extent from the limited knowledge of these systems and details on precisely how they function. The mineral created may in fact be todorokite (i.e. delatorreite), which is a more complex manganese oxide (Cravotta, 1999). The microorganisms that oxidize the metals may be inherent in nature. Therefore, culturing and inoculation procedures may not be necessary. There are size considerations in the construction of these systems due to the relatively long residence times recommended (2.5 to 3.0 days). A large flow rate would require a fairly large system for successful treatment. It is also uncertain how highly acidic ($pH < 4.0$) metal-laden water would affect the treatment process.

Alkalinity-Producing Diversion Wells

Alkalinity-producing diversion wells, a low maintenance method for treating acidic water, were developed in Norway and Sweden using a water pressure-driven, fluidized limestone bed. This technology has been modified for use in treating AMD and streams contaminated by AMD (Arnold, 1991).

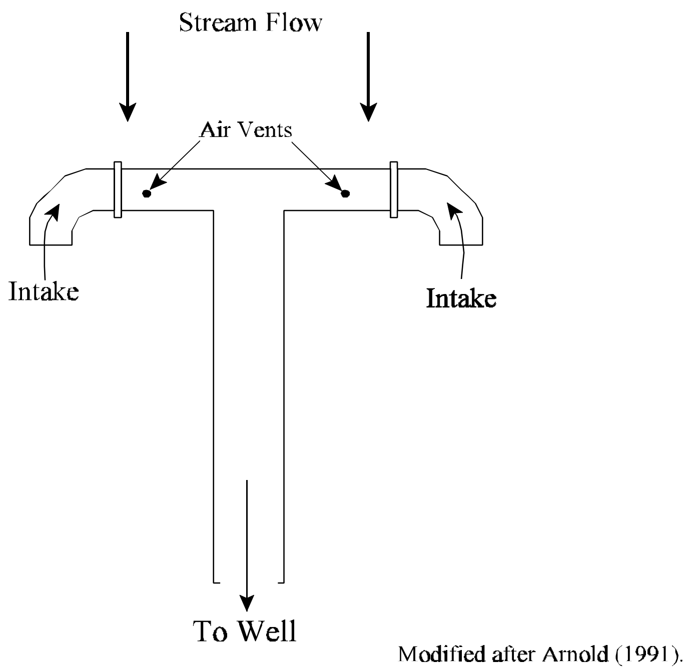
Typically, these diversion wells are large cylinders (commonly 5 to 6 feet in diameter and 6 to 8 feet high) composed of reinforced concrete or other erosion resistant material (Figure 4.1e). Two manhole sections, one on top of the other, are frequently used. The bottom of the well should be equally strong and erosion resistant and is commonly formed from reinforced concrete. Water is piped into the center of the well with the end of the pipe just above the well bottom (2 to 3 inches). The outlet point can also be fitted with a metal collar with holes drilled in the sides. This will direct the water sideways and appears to be more efficient than directing the water downward. An 8 or 10 inch pipe size is recommended to provide the required flow rate. The water is fed from a point up gradient, where the water is dammed to yield a consistent 8 feet of head above the well surface (Arnold, 1991). A driving head of 10-12 feet was suggested by McClintock and others (1993). Only a portion of the stream flow is diverted, while the rest continues to flow normally downgradient. The recommended flow rate should average about 2,244 gpm (Arnold, 1991), however, observations of working wells in eastern Pennsylvania, indicate that a flow rate of 112 to 224 gpm may sufficiently operate diversion wells. McClintock and others (1993) stated that stream flows as low as 100 gpm can be treated with diversion wells. At low-flow streams virtually all of the flow will be routed through the well. Crushed limestone is dumped into the well. The optimum size of the limestone is one-half to three-quarters of an inch. Smaller size particles tend to easily wash out and larger particle sizes require higher flow rates to maintain a fluidized bed. The rapid upward movement of the water through the well causes the limestone chips to roil creating a fluidized bed. The top of the well is flared to accommodate an energy reduction in the upward flow which inhibits limestone from washing out. The well is maintained to be consistently approximately half full of limestone (Arnold, 1991).

Figure 4.1e: Typical Alkalinity-Producing Diversion Wells



The water intake point needs to be constructed to inhibit the uptake of leaves, sticks, and other debris, which tend to clog the plumbing. Arnold (1991) recommends a tee with each side fitted with an elbow open toward downstream (Figure 4.1f). Air vents drilled into the tee are recommended to allow the bleeding off of entrained air from vortex action and from air entrained during low flow periods (Arnold, 1991).

Figure 4.1f: Example of a Water Intake Portion of an Alkalinity-Producing Diversion Well



These wells yield alkalinity from acidic water that reacts directly with the limestone and by the churning action of the fluidized bed grinding the limestone into fine particles. The finer limestone particles will also react with the water in the well, imparting additional alkalinity and are carried out of the well and to the stream to react with the remaining acidic water that is not piped through the well. The constant churning and surface abrasion of the limestone prevents armoring by dissolved iron in the mine drainage. Limestone consumption rates vary with flow rate, well size, limestone purity and hardness, and to a lesser extent water quality. However, these wells are generally designed to use approximately 0.92 yd^3 of limestone per week. Purer limestones are recommended, because highly dolomitic, very hard limestones tend to react too slowly (Arnold, 1991). It is important to note limestones that are too soft will break up too easily, rapidly wash out of the well, and require more frequent replenishment.

The turbulent action within the wells preclude *in-situ* iron deposition. Any dissolved iron present, above 0.3 mg/L, will likely precipitate after leaving the well. It may be prudent to have a settling pond constructed between the well and the receiving stream to collect much of the precipitating iron and other metals.

Arnold (1991) recorded an increase of one to two pH units (orders of magnitude) of the water leaving the diversion well at 5 cfs. McClintock and others reported a pH increase of up to 3 orders of magnitude. Arnold anticipated a rise in alkalinity proportional to the pH increase, and which alkalinity was increased somewhat, but the concentrations remained relatively low. No detrimental impacts on the in-stream aquatic life were noted with the use of diversion wells (Arnold, 1991). The limited alkalinity production is due primarily to the low (atmospheric) levels of CO_2, which govern the rate of limestone dissolution. Watten and Schwartz (1996) proposed pretreating the mine water by injecting CO_2 under pressure (100 psi), which increases CO_2 saturation by 22,000 fold. This CO_2 saturation increases the potential alkalinity production to 1,000 mg/L (Watten and Schwartz, 1996). However, CO_2 injection is not passive in nature and would dramatically increase the cost and labor of the operation.

There are some restrictions in the use of diversion wells. These include, but are not limited to:

- Sufficient grade is required to maintain the 8 to 12 feet of head.
- Sufficient flow is required to keep the well functioning properly.
- Waters with high acidity concentrations will not be completely treated by one pass through a well. The water may need to be piped through a battery of wells to achieve complete neutralization.
- There is more maintenance required for these wells than is needed for other passive treatment systems. Recharging of the limestone may need to be performed on a weekly basis.
- If considerable dissolved iron is present, an additional settling pond may be required.
- Intake clogging may be a problem during certain times of the year. Keeping the intake clear and unclogging of the entire piping system are periodic maintenance requirements.

Design Criteria

Passive treatment systems are designed to inexpensively treat AMD with very little to no maintenance once constructed. These systems are engineered to raise the alkalinity and pH while facilitating the precipitation of metals. The mechanisms of AMD treatment rely on metals oxidation or reduction and the production of alkalinity by sulfate reduction or limestone dissolution. The design of these treatment systems varies according to the type, but there are some basic requirements that are common to all. The following list includes basic criteria of passive treatment systems:

- Data are required to determine anticipated flow rates and water quality.
- The size of the facility is based to a large extent on flow rates and detention time.
- The type of system to be employed is directly dependent on water quality (e.g., pH, ferrous vs ferric iron, dissolved oxygen content, net alkalinity, etc.).
- The highest $CaCO_3$ purity limestone is recommended.
- Considerable area is generally required to construct these systems.
- Sufficient grade is required to permit gravity-driven water flow through these systems.
- Flow through these systems needs to be consistent. An interruption of flow can cause the treatment efficacy to be compromised.
- ALDs require low levels of dissolved oxygen, dissolved iron to be virtually all ferrous, and low levels of dissolved aluminum.
- Aerobic wetlands work best when the pH is elevated and there is a net alkalinity.
- To maintain efficiency, SAPS, oxic limestone drains, and open limestone channels require periodic flushing to wash out the loose metal precipitates.

4.2 Verification of Success or Failure

As with all BMPs, verification of proper implementation is crucial to effective control or remediation of the discharge pollution loadings. Monitoring of the water quality and quantity will be the truest measure of the effectiveness of these BMPs. The importance of field verification of all aspects of a BMP cannot be overstated. It is the role of the inspection staff to enforce the provisions outlined in the permit. The inspector generally does not need to be present at all times to assess the implementation of the BMPs in this chapter. However, during installation, some passive BMPs will require closer and more frequent field reviews than others.

The truest test of the success of passive treatment is the water quality of the effluent compared to the influent. This assessment is determined through sampling and analysis of the water and measurement of the flow rate. A sampling and measurement port is needed to access the discharge prior to treatment. An assessment of influent verses effluent flow rates is also necessary. Greater outflow than inflow is indicative that the system is gaining unaccounted-for water within the system. If the outflow is less than the inflow, the system is likely leaking. If the treatment system is gaining or losing unaccounted-for water, it should be repaired. Topographic maps or surveying can be used to determine if sufficient grade exists to adequately drive the flow of these systems.

Implementation Checklist

There are several items that should be monitored to ensure these treatment systems are adequately engineered and installed. This list includes but is not limited to:

- Measurement of flow rate and analysis of the water quality of the discharge. Treatment system engineering is based on these data. Water should be especially analyzed for DO, ferrous and ferric iron, acidity, pH, alkalinity, dissolved aluminum, and dissolved manganese.
- Measurement of the flow rate and analysis of the water quality of the system effluent. Compare effluent quality to raw water for efficiency determinations.
- Monitor the amounts, size, and purity of any limestone used. Limestone purity should be determined from laboratory analysis. Monitor the type and amount of organic materials. The amount of limestone can be determined from reviewing the weigh slips or estimated from the stockpile dimensions.

- Review background data, especially flow, iron concentration, and acidity concentration, to determine the adequate sizing of the treatment systems.
- Monitor crucial portions of the system installation.
- Check for unwanted water infiltration and/or leaks.
- Determine if sufficient grade exists to create the head required to run these systems.

Many of the verification techniques are common to several passive treatment types, while others may be system-specific. The following list include implementation verification techniques for passive treatment systems:

<u>ALDs</u>

- The size of the trench can be measured during excavation for comparison to the calculated amount of crushed limestone required for treatment. A cubic yard of crushed limestone (1.5 to 2.0 inch) weighs about 2,300 pounds (Nichols, 1976).
- Cover material (e.g., plastic and clays) can be inspected prior to use or can be viewed during installation. If there is a concern as to the adequacy of this material, certification of the strength, permeability, and other properties can be required.
- The DO and/or iron oxidation state of the effluent can be analyzed to ascertain the ability of the drain to preclude atmospheric oxygen.
- A lack of drain outflow and/or the existence of unanticipated discharge points are indicative that the drain is clogged and/or cannot handle the amount of water piped into it.
- Drains should be sized to permit at least a 15 hour, preferably 23 hour, detention time.

<u>Constructed Wetlands</u>

- Sizing of wetlands can be directly measured and compared to the flow rate to determine if they were sized adequately to properly treat the water. It is recommended to use a sizing factor of 10 gdm for water with a pH of greater than 4.0 and 4 gdm if the pH is less than 3.0 (Hedin and Nairn, 1990). However, a sizing factor of 15 gdm may provide reasonable results (Kepler, 1990).

- the optimal flow through the wetland can be determined from visual observation or by use of tracing dyes .
- Lack of vegetation may be an indication that the water level is too high or too low.

SAPS

- The size of the system can be measured during excavation for comparison to the calculated amount of crushed limestone required for treatment.
- Sizing of SAPS can be directly measured and compared to the flow rate, (using the above referenced sizing criteria) to determine if it is adequate for proper treatment.
- Effluent water quality can be monitored to determine if the iron is being reduced and the DO is being removed.
- The water level should be monitored to ensure that the SAPS will not be dewatered or overflow. Either situation will impede the effectiveness of the system.
- SAPS should be sized to permit a detention time similar to ALDs (15 to 23 hours).

Open Limestone Channels

- The size of the trench can be measured during excavation and compared to the calculated amount of crushed limestone required for treatment.
- Sizing of channels can be directly measured and compared to the flow, using the above referenced sizing criteria, to determine if it is adequate for proper treatment.
- Visual inspection or inadequate flow rate will indicate if the metal floc is clogging the pore spaces in the limestone.
- Flow-through rate and average detention time can be determined by use of dye tracing.
- Recommended detention time is at least 3 hours to effect 100 percent acidity neutralization.

Oxic Limestone Drains

- The size of the trench can be measured during excavation and compared to the calculated amount of crushed limestone required for treatment.

- Proper sizing of drains can be directly measured and compared to the flow, using the above-referenced sizing criteria, to determine if it is adequate for proper treatment.
- A lack of outflow and/or unanticipated discharge points are indicative that the drain is clogged and/or cannot handle the amount of water piped into it.
- Drain residence times of $\leq$3.1 hours and water velocities of 0.33 to 1.31 feet per minute are adequate to effect treatment and flush out the metal flocs.
- Flow through rate and average detention time can be determined by use of dye tracing.

<u>The Pyrolusite® Process</u>

- The size of the trench can be measured during excavation and compared to the calculated amount of crushed limestone required for treatment.
- Sizing of beds can be directly measured and compared to the flow, using the above referenced sizing criteria, to determine if it is adequate for proper treatment.
- A minimum detention time of 2.5 to 3.0 days is recommended.

<u>Alkalinity-Producing Diversion Wells</u>

- The size of the well can be measured during excavation.
- Sizing of well can be directly measured and compared to the flow, using the above-referenced sizing criteria, to determine if it is adequate for proper treatment.
- The in-stream improvement as well as the quality of the well effluent are indicative of the efficiency of these systems.
- A head of 10 to 12 feet is required to run the system. Flow rates of 100 gpm to over 2,000 gpm can be treated.

4.3 Case Studies

<u>Case Study 1</u> (Appendix A, EPA Remining Database, 1999 TN (5))

This site is located in Campbell County, Tennessee, approximately 4 miles north of Caryville. The operation was permitted for 201 acres adjacent to Interstate 75 with roughly 108 acres of coal removal. This was a conventional SMCRA permit, for non Rahall-type remining, and accessed the Coal Creek coal seam. Passive treatment was used effectively to treat the post-mining effluent. Problems arose at this site when operations were ceased, due to a fatal fly rock incident from blasting of the overburden. After approximately 80 percent of the mining had been completed, the operation was ceased and never reactivated. The performance bonds were eventually forfeited and a mine drainage problem developed from flooding of the pit, lack of proper handling of acid-forming materials, no contemporaneous reclamation, and other undesirable conditions.

In order to remediate the problem, the Tennessee Valley Authority (TVA), owner of the mineral rights, undertook the task of reclaiming the site and installed a series of passive treatment systems to treat the water. They elected to install an ALD followed by staged aerobic wetlands.

An underdrain was installed across the pit floor as part of the mining process. The outflow of the underdrain was intercepted and an ALD was tied into it. The ALD was designed for a 30 year lifespan with almost 3,200 tons of limestone used. Prior to entering the drain, the discharge was slightly net alkaline (~50 mg/L), with around 40 mg/L dissolved ferrous iron, and an expected flow estimated at 160 gpm. The drain was designed to yield 250 mg/L alkalinity.

The discharge of the ALD was piped to the staged wetlands. The wetlands were designed to remove 20 gdm of iron and 0.5 gdm of manganese. Based on these removal rates, the wetlands were sized at 3.45 acres. Initially, the ALD effluent was piped to an oxidation pond to permit primary treatment (abiotic oxidation of metals, hydrolysis, and subsequent precipitation) and to prolong the effective life of the wetland. The pond was 0.77 acres with a detention time of about 24 hours. Following the pond, the mine water flowed into a 2.7 acre wetland. The wetland was divided into five cells with different water levels and vegetation. The first cell had an average of 3 feet of water and was planted with rice cutgrass, wool grass, and arrowhead. The area of the first cell was 1.02 acres. The second cell had an average of 18 inches of water over 0.59 acres and was planted with cattail, rice cutgrass, and bulrush. The third cell was 0.44 acres with an average water depth of 8 inches and was planted with wool grass, arrowhead, and burreed. The fourth cell was 0.35 acres with an average of 10 inches of water and planted with wool grass, arrowhead, bulrush, burreed, and sedge. The last cell was 0.3 acres with an average depth of 12 inches of water and was planted with cattails. Following the last wetland cell, the water was channeled to an existing basin for final polishing prior to discharging.

The water of the underdrain discharge prior to the ALD installation (given by the TVA) had a pH of 6.0, 40 mg/L iron, 7 mg/L manganese, 15 mg/L acidity, and 65 mg/L alkalinity. The flow was given as 160 gpm. These values were used for treatment system design criteria. Once the passive system was installed, the raw discharge water could no longer be sampled. Table 4.3a is a summary of the water quality at various points as it flows through the treatment system from November 1996 through August 1998.

Table 4.3: Summary of Water Quality Data at Various Points Along a Passive Treatment System

Sample Point	Median Flow (gpm)	Median pH (Standard Units)	Median Alkalinity Concentration (mg/L)	Median Iron Concentration (mg/L)	Median Manganese Concentration (mg/L)
ALD Effluent	186.5	6.2	196	59.50	24.8
Fourth Wetland Effluent	197.5	6.9	106	0.88	22.6
Last Settling Pond Effluent	197.0	7.0	100	0.82	11.1

It appears that initial flow estimates used in sizing the system were too low. The median flow through the system was about 23 percent above the pre-installation estimate. However, the system has effectively raised the alkalinity. The alkalinity after the ALD is over three times greater than the underdrain inflow value. The alkalinity is lowered as the water flows through the wetland by release of mineral acidity as iron and manganese are oxidized and hydrolyze. The final effluent alkalinity remains over 50 percent above the levels exhibited by the underdrain. The final pH (~7.0) is significantly above the pH of the ALD influent (~6.0). Iron concentrations are dramatically reduced from near 60 mg/L to well below BAT effluent standards (<1.0 mg/L).

Manganese is reduced by greater than 50 percent, but continues to be well above effluent standards. The continued manganese problem may be due to the apparent undersizing of the system. It is uncertain how the 160 gpm was determined for the discharge prior to sizing the treatment system. Analysis of the existing data indicates that the median flow prior to installation of the treatment system was nearly 190 gpm.

4.4 Discussion

The remining Best Management Practice discussed in this section relates to improvement of effluent by end-of-the-pipe treatment of mine water. Because these systems can be considered as treatment of mine water, they may not necessarily be categorized as true BMPs. There are exceptions where a passive treatment technology or system may qualify as an integral BMP. If an ALD is incorporated within the backfill as a pit floor drain, it can be considered a traditional BMP. If a passive treatment system is installed to treat a discharge that is adjacent to the remining operation and outside of the permit boundary, but is not hydrologically connected to the operation, this also could be considered a BMP. In other words, the operator installs passive treatment on an adjacent discharge, not legally associated with the remining site, to improve the overall watershed water quality.

Benefits

- Low maintenance method to reduce the pollution load of mine water.
- Means of gaining additional water quality improvement on and above what is capable with traditional BMPs.
- Some systems are capable of yielding very high amounts of alkalinity and thus, additional buffering capacity, by maintaining elevated CO_2 concentrations.

Limitations

- Generally require a substantial construction area for moderate to high-flow discharges.
- Require topography that provides sufficient gradient for gravimetric flow.

- Need to be refurbished periodically for cleaning out or replenishment of the reactive materials.
- Certain water quality parameters (e.g., ferric iron, aluminum, or low pH) can cause some systems to fail or to perform below peak efficiency.
- Metals removal and alkalinity are limited by detention times and chemical reaction rates.

Efficiency

Very few of completed remining sites in Pennsylvania (Appendix B: PA Remining Site Study, 1999) utilized passive treatment as an integral part of their BMP plan. In this study, 2 out of a total of 231 discharges were effected by passive treatment BMPs. However, only one discharge was treated with a passive treatment BMP for a manganese problem. A statistical evaluation of these data is not powerful, because of the extremely limited data. However, no discharge exhibited significantly degraded water quality for acidity, iron, manganese, or aluminum loadings. One discharge was significantly improved for acidity, iron, manganese, and aluminum loadings. The other discharge was unchanged for acidity, iron, and aluminum loadings.

Additional remining sites are required to conclusively evaluate the use of passive treatment BMPs in improving effluent pollution loads. However, the research into passive treatment indicates that in most cases a water quality improvement can be anticipated.

4.5 Summary

Passive treatment technology, although not generally a traditional BMP, can be used to augment pollution load reduction achieved by implementation of true BMPs. Passive treatment provides low cost and minimum labor methods to treat AMD for acidity and certain metals. Research into passive treatment illustrates that a variety of systems can be used to treat a broad range in water quality. The type of systems to be employed should be tailored specifically to the mine water quality.

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